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DOSE CONVERSION FACTORS DERIVED FOR POPULATION RESIDING IN THE GROUND WATER DISCHARGE AREA CONTIGUOUS TO RW GEOLOGICAL REPOSITORY

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Simplified conservative method was applied to assess biosphere dose conversion factors for a scenario suggesting the use of contaminated water for household and drinking water supply needs, including irrigation of eatable plants. Calculations show that the main contribution to radiation dose is associated with the consumption of vegetables irrigated by contaminated water with root uptake being considered as predominant contamination path.

Key words: *radionuclides, radioactive waste, underground disposal, ingestion dose, critical group of population.*

Introduction

Effective dose or the radiation risk being linearly coupled to it is considered as a quantitative criterion for the safety assessment of any radiation hazardous facility. In particular, this is also relevant for deep radioactive waste disposal facilities (DRWDF) with the demonstration of their long-term safety for the entire period of the RW potential hazard being considered as a key and most pressing issue addressed during their construction (or even renouncing the idea of their construction) [1]. Since radionuclides posing hazard for millions of years are present in DRWDF, processes associated with changing climate, landscape, behavioral preferences of the population, etc. should be accounted for in such safety demonstrations. Reliable forecasting given such long timeframes seems to be quite problematic, since the maximum forecasting timeframe providing for more or less adequate quantitative estimates is currently believed to be no more than 10,000 years [2].

Estimates covering a time period until the radionuclide release and radiation exposure on the biosphere reaches its maximum values is seen as an alternative method allowing to make relevant forecasts for millions of years. Under such an approach, relevant task for a DRWDF located in crystalline rocks is somewhat facilitated, since maximum release of radionuclides occurs within a time interval (several thousand years [3—4]), during which external conditions and characteristics of safety barriers remain relatively constant and can be quantitatively characterized.

All sorts of internal and external influences and related processes are taken into account under safety assessments as part of developed scenarios with the initial or basic one being specified at their preliminary stage. This scenario constitutes the first step and the basis of DRWDF safety assessments. Among its features, conservatism and the assumption suggesting the constant nature of climate conditions and daily living activities of populations being potentially exposed to the radiation impacts associated with the DRWDF are considered of primary importance. The second assumption seems to be quite reasonable, since recent studies [5–6] indicate that for the next 100,000 years or so the currently existing climatic setup will not suffer any significant changes[1].

Currently, long-term safety assessments are being actively performed for the DRWDF developed in Russia: construction activities were launched in the Krasnoyarsk Region (in the vicinity of Zheleznogorsk city) with their first stage involving the construction of an underground research laboratory [7—9]. DRWDF's safety demonstration is seen as a long iterative process embracing the entire life cycle of the facility with the estimates obtained to be improved as more information both about the facility itself and about its environment is gained and updated, including the mechanisms accounting for the potential exposure of personnel and the population.

This study is focused on demonstrating how the initial data are obtained for the integral models. The main task of such models is to perform simplified conservative assessments of the radiological impact on the population and the environment under a setup suggesting a shortage of initial data. As new information is received, models should become more complex and realistic.

Based on this study, conversion coefficients allowing to establish a relationship between the activity of a radionuclide flux from the DRWDF into the biosphere (Bq/year) and the expected dose exposure (Sv/year) were derived. Conversion coefficients are given in Sv/Bq units.

The paper focuses on a scenario suggesting that contaminated groundwater that was in contact with the DRWDF flows into a well the water from which is used for drinking supply, cattle watering and irrigation of gardens where products for own consumption are grown (Figure 1).

Population

A settlement was chosen from [10] as a prototype for the initial data selection (mainly based on dietary structures). Production grown by private farm households (vegetables, meat, milk), wild plants (mushrooms, berries) and fish caught in the river make up the main food source for its residents. Table 1 shows the average diets considered in this

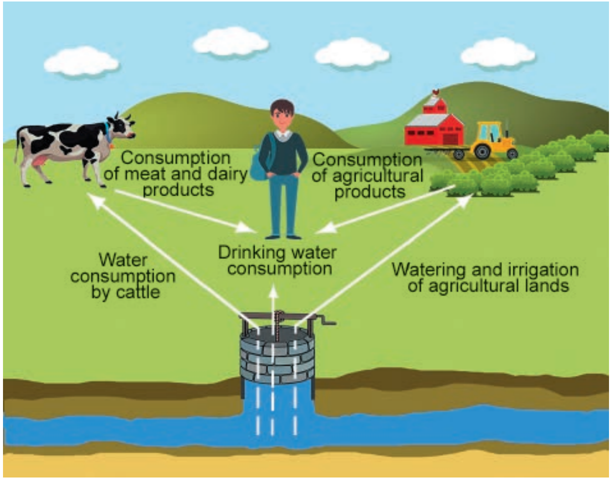

Figure 1. Illustration of the calculation model

paper. Solely products being potentially susceptible to contamination due to the use of water from the well were taken into account. To evaluate the kids' diet, adult/child ratios from [11] providing for the main provisions of the methodology were applied. It should be noted that the consumption values used for vegetables, meat and milk are about 2 times lower than the average European ones [11]. It was assumed that this diet refers specifically to local products potentially containing radionuclides released from the DRWDF. Missing products for proper nutrition were assumed to be imported and were not considered in the calculations.

Table 1. Average dietary intake of critical populations adopted in the calculations, kg (l)/year

Critical group	Water	Vegetables (potatoes)	Meat	Milk
Adults	600	182	59	136
Children	260	67	24	163

Radionuclides and contaminated groundwater

Main dose-contributing radionuclides determining the potential radiation impact of the DRWDF at the post-closure stage and following the expiration of administrative control period were considered in the performed calculations. The following radionuclides were considered in further analysis given their potential rising to the surface in a liquid phase: ^{14}C, ^{36}Cl, ^{79}Se, ^{93}Mo, ^{94}Nb, ^{99}Tc, ^{129}I, ^{135}Cs, ^{226}Ra, ^{235}U, ^{238}U. These radionuclides account for more than 99 % of the DRWDF potential radiation impact on the population [3—4]. Table 2 presents the decay constants of the listed radionuclides $\lambda_i = \ln(2)/T_{1/2,i}$ ($T_{1/2,i}$ is the half-life of the i-th radionuclide) and dose coefficients for food intake accounting for the selected critical groups based on [12].

[1] Perhaps a slight warming due to the greenhouse effect is possible, however, there no univocal opinion on this issue has yet emerged.

Table 2. Decay constants and dose coefficients for radionuclides accounting the ingestion intake

Nuclide	$T_{1/2}$, years	λ_i, year^{-1}	Dose coefficients, Sv/Bq	
			Adults*	Children
^{14}C	$5.73 \cdot 10^{3}$	$1.21 \cdot 10^{-4}$	$5.8 \cdot 10^{-10}$	$1.6 \cdot 10^{-9}$
^{36}Cl	$3.01 \cdot 10^{5}$	$2.30 \cdot 10^{-6}$	$9.3 \cdot 10^{-10}$	$6.3 \cdot 10^{-9}$
^{79}Se	$6.50 \cdot 10^{4}$	$1.07 \cdot 10^{-5}$	$2.9 \cdot 10^{-9}$	$2.8 \cdot 10^{-8}$
^{93}Mo	$3.50 \cdot 10^{3}$	$1.98 \cdot 10^{-4}$	$3.1 \cdot 10^{-9}$	$6.9 \cdot 10^{-9}$
^{94}Nb	$2.03 \cdot 10^{4}$	$3.41 \cdot 10^{-5}$	$1.7 \cdot 10^{-9}$	$9.7 \cdot 10^{-9}$
^{99}Tc	$2.13 \cdot 10^{5}$	$3.25 \cdot 10^{-6}$	$6.4 \cdot 10^{-10}$	$4.8 \cdot 10^{-9}$
^{129}I	$1.57 \cdot 10^{7}$	$4.41 \cdot 10^{-8}$	$1.1 \cdot 10^{-7}$	$1.9 \cdot 10^{-7}$
^{135}Cs	$2.30 \cdot 10^{6}$	$3.01 \cdot 10^{-7}$	$2.0 \cdot 10^{-9}$	$2.0 \cdot 10^{-9}$
^{226}Ra	$1.60 \cdot 10^{3}$	$4.33 \cdot 10^{-4}$	$2.8 \cdot 10^{-7}$	$1.5 \cdot 10^{-6}$
^{235}U	$7.04 \cdot 10^{8}$	$9.85 \cdot 10^{-10}$	$4.7 \cdot 10^{-8}$	$1.3 \cdot 10^{-7}$
^{238}U	$4.47 \cdot 10^{9}$	$1.55 \cdot 10^{-10}$	$4.5 \cdot 10^{-8}$	$1.2 \cdot 10^{-7}$

* In [12] ^{79}Se dose coefficient for adults is missing. The missing value was taken from [15].

It was assumed that the plume of contaminated groundwater enters the upper aquifer used for drinking and irrigation purposes. Thickness of this horizon was assumed as $h = 30$ m with its filtration coefficient $K = 0.1$ m/day and the hydraulic head gradient $i = 0.023$ [13, 14]. It was also assumed that the surface area of the radioactive plume when entering the surface aquifer corresponds to the surface area of the settlement ($6 \cdot 10^5$ m^2) and its linear size accounts for $L = \sqrt{6 \cdot 10^5} \approx 775$ m. Groundwater flow in this aquifer under the settlement (Q) can be calculated using the Darcy law:

$$Q = L \cdot h \cdot v_{\mathrm{Darcy}} = L \cdot h \cdot K \cdot i \approx 2 \cdot 10^4 \text{ [m}^3\text{/year]}. \qquad (1)$$

It was assumed that the flow rate of contaminated groundwater is negligible compared to the surface flow (there is no dilution of the groundwater surface flow) and for a single radioactivity flow (1 Bq/year) the concentration of radionuclide C_w [Bq/m^3] in the aquifer and, therefore , in the well water of the settlement will be equal to $1/Q \approx 5 \cdot 10^{-5}$ Bq/m^3.

Contamination of plant products due to irrigation with water containing radionuclides

The calculations are based on a methodology allowing to estimate emissions and discharges of radionuclides under individual and collective doses assessments described in [11]. These models were developed to determine the expected radiation exposure from nuclear facilities based on simplified conservative estimates. This paper presents a methodology for operational dose assessments using minimum information being specific to a particular nuclear site. In order to simplify the models, a number of processes are considered using complex parameters describing the effects of several interrelated processes.

In [11], the concept of control levels (CL) was set forth — these are commonly specified as being several times lower than the standard dose limits. If preliminary conservative estimates show that the calculated values do not exceed the CL, then further assessment is not required. If exceedance is revealed, more realistic assessments involving more complex models are required.

Given specific features of such facilities as DRWDF associated with huge uncertainties in relevant long-term safety assessments and with the results not exceeding the regulatory limits and accounting for a large margin of safety being considered acceptable, the most conservative approach was chosen for conversion coefficient identification [11].

It was believed that the exposure of plants to radionuclides and their retention on plants occurs as a result of irrigation with contaminated water or secondary dust elevation. Radionuclides reaching the plants surface or absorbed by their roots from the contaminated soil can be allocated in their tissues. A number of processes can potentially lead to a decrease in the specific activity of nuclides contained in the plants, including radioactive decay, a decrease in concentration due to the growth (weight gain) of plants, radionuclides washing off from the surface, leaching and tillage. Further removal of radionuclides from the plants can occur during cattle grazing, harvesting, etc. Under simplified estimates, all these processes are taken into account using a single empirical coefficient in an exponential dependence (2).

To calculate the concentration of radionuclides $C_{v,i,1}$ due to direct contamination with radionuclide i during irrigation, the following equation was used:

$$C_{v,i,1} = \frac{\dot{d}_i \alpha \left[1 - \exp(-\lambda_{E_i^v} t_e \right]}{\lambda_{E_i^v}}, \qquad (2)$$

where $C_{v,i,1}$ is the concentration of the i-th radionuclide in plants consumed by humans measured in [Bq/kg] of fresh weight; $\dot{d}_i$ [Bq/(m^2·year)] is the intensity of the i-th radionuclide release during irrigation per unit area; α [m^2/kg] is the coefficient specifying part of the activity captured by the edible part of plants per unit mass (coefficient of mass capture = 0.3 m^2/kg for edible plants) with the fresh weight applied for fresh edible plants. $\lambda_{E_i^v}$ [year^{-1}] is the effective temporary constant indicating the decrease in the specific activity of the i-th radionuclide being equal to $\lambda_i + \lambda_w$; t_e [year] is the irrigation time (assumed as 120 days during the season or 0.33 year^{-1}); λ_w [year^{-1}] is the temporary constant indicating the decreasing activity in the plant due to all processes except for radioactive decay (assumed to be equal to 0.05 day^{-1} or 18 year^{-1}); λ_i [year^{-1}] is the decay constant of the i-th radionuclide.

The following equation was used to calculate the concentration of radionuclides in plants associated with indirect processes — root uptake and exposure to contaminated soil particles:

$$C_{v,i,2} = F_v \cdot C_{s,i}, \tag{3}$$

where $C_{v,i,1}$ is the i-th radionuclide concentration in plants consumed by humans measured in [Bq/kg] of fresh weight; F_v is a factor being specific to chemical elements and indicating the concentration of radionuclides captured from the soil and accumulated in edible parts of plants [Bq/kg of the plant's fresh weight per Bq/kg of dry soil]. This factor also takes into account the contaminated soil particles being captured by plants; $C_{s,i}$ [Bq/kg] is the concentration of the i-th radionuclide in dry soil, which can be derived from the following ratio:

$$C_{s,i} = \frac{\dot{d}_i \left[1 - \exp(-\lambda_{E_i^s} t_b)\right]}{\rho \lambda_{E_i^s}}, \tag{4}$$

where $\lambda_{E_i^s}$ [year^{-1}] is the effective temporary constant indicating the decreasing concentration of the i-th radionuclide in the soil within the plant's root system zone being equal to $\lambda_i + \lambda_s$; λ_s [year^{-1}] is the temporary constant indicating the decreasing activity in the soil within the plant's root system zone due to all processes except for the radioactive decay; λ_i [year^{-1}] is the radioactive decay constant of the i-th radionuclide; t_b [year] is the time period for radionuclide release into soil (assumed as being equal to 10^4 years); ρ [kg/m^2] is the surface density of the soil within the plant's root system.

Soil contamination due to irrigation

It was assumed that during irrigation, a total of 9 l/m^2 of water was poured per day and the irrigation itself was carried out for 16 days (once a week for 4 months). The average irrigation intensity at this time was calculated as follows:

$$I_w = 9\ \text{l}\cdot\text{m}^2\cdot\text{day}^{-1} \times 16\ \text{day/year} \times 10^{-3}\ \text{m}^3/\text{l} = 0{,}144\ \text{m/year}. \tag{5}$$

Given that the concentration of the i-th radionuclide in water used for irrigation $C_{w,i}$, accounts for $5 \cdot 10^{-5}$ Bq/m^3 for a single activity flux (see above), the intensity of the i-th radionuclide release into soil and vegetation can be calculated as follows:

$$\dot{d}_i = I_w \cdot C_{wi} = 0{,}144\ \text{m/year} \times 5 \cdot 10^{-5}\ \text{Bq/m}^3 = 7{,}2 \cdot 10^{-6}\ \text{Bq/(m}^2\cdot\text{year)}. \tag{6}$$

Decreasing radionuclide concentrations in soil

The initial concentration of radionuclides released into soil may decrease subsequently due to soil erosion, mixing of contaminated and non-contaminated soil, plowing, surface runoff, downward movement of radionuclides due to leaching, as well as due to radioactive decay. Binding of radionuclides in the soil particle matrix is viewed as an important process which seems to be especially important, for example, for Cs. Concentration of radionuclides in the soil can be reduced due to root uptake and subsequent removal of plants during harvesting or eating by animals. For typical models, all of the above processes, except for radioactive decay, are described by a single time constant λ_s [year^{-1}]. In real-life situations, λ_s depends significantly on the climate, agricultural methods, types of soils, vegetation and chemical forms of radionuclides. In the described approach, anionic forms of nuclides, Cs isotopes and other nuclides are distinguished. Anions such as TcO_4^-, Cl^-, I^- leach out quickly with relevant characteristic value of λ_s amounting to 0.5 year^{-1}. For Cs, λ_s is assumed to be equal to 0.05 year^{-1}, for other nuclides (including non-anionic forms of Tc), the default values are assumed to be equal to 0.

Specific aspects of radioactive fallout accounting in the model

Fallouts on the soil are possible from the atmosphere and during irrigation. If foliage is available, radionuclides can be released into the soil due to leaf fall, leaching (from plants), washing off dead tissue, with excrements or with moving pasture-fed animals. For typical models, it is conservatively assumed that all fallouts enter the soil regardless of their possible capture by plants' leaves and harvesting. Thus, using irrigation intensity formula (4), one can significantly overestimate the radionuclide uptake by soil. This is especially true for radionuclides being present in anion form. Dedicated experts should be engaged in such studies to provide more realistic estimates.

Radionuclide concentration in the soil also depends on the average soil depth, over which the fallout activity is being averaged, and its density, which in turn depends on the soil type and compaction. $\rho = 260$ kg/m^2 corresponding to the root zone depth of food plants being equal to 20 cm and the soil density of 1,300 kg/m^3 was assumed under this approach.

Total radionuclide concentration in plants

Total concentration of a radionuclide in plants at the time of their consumption by animals or people is calculated as follows:

$$C_{v,i} = (C_{v,i1} + C_{v,i2})\exp(-\lambda_i t_h), \qquad (7)$$

where $C_{v,i}$ [Bq/kg] is the concentration of i-th radionuclide per kg of fresh weight for plants consumed by humans; λ_i [year^{-1}] is the radioactive decay constant of the i-th radionuclide; t_h [year] is the time interval between harvesting and plant consumption (taken equal to 14 days or 0.038 years).

Table 3 shows the F_v concentration factors and the expected radionuclide concentrations in plants for a single radionuclide activity flux from the DRWDF calculated according to (1)—(7).

Table 3. Concentration factors and expected radionuclide concentrations in plants

Nuclide	F_v*	$C_{v,i1}$ (Bq/kg)	$C_{v,i2}$(Bq/kg)	$C_{v,i}$ (Bq/kg)
^{14}C	0.1	$1.2\cdot10^{-7}$	$1.6\cdot10^{-5}$	$1.6\cdot10^{-5}$
^{36}Cl	40	$1.2\cdot10^{-7}$	$2.2\cdot10^{-6}$	$2.3\cdot10^{-6}$
^{79}Se	0.1	$1.2\cdot10^{-7}$	$2.6\cdot10^{-5}$	$2.6\cdot10^{-5}$
^{93}Mo	0.2	$1.2\cdot10^{-7}$	$2.4\cdot10^{-5}$	$2.4\cdot10^{-5}$
^{94}Nb	0.01	$1.2\cdot10^{-7}$	$2.3\cdot10^{-6}$	$2.5\cdot10^{-6}$
^{99}Tc	5	$1.2\cdot10^{-7}$	$2.8\cdot10^{-7}$	$3.9\cdot10^{-7}$
^{129}I	0.02	$1.2\cdot10^{-7}$	$1.1\cdot10^{-9}$	$1.2\cdot10^{-7}$
^{135}Cs	0.3	$1.2\cdot10^{-7}$	$1.7\cdot10^{-7}$	$2.8\cdot10^{-7}$
^{226}Ra	0.04	$1.2\cdot10^{-7}$	$2.5\cdot10^{-6}$	$2.6\cdot10^{-6}$
^{235}U	0.01	$1.2\cdot10^{-7}$	$2.8\cdot10^{-6}$	$2.9\cdot10^{-6}$
^{238}U	0.01	$1.2\cdot10^{-7}$	$2.8\cdot10^{-6}$	$2.9\cdot10^{-6}$

* No F_v values were indicated in [11] for C and Cl. Values for carbon were taken from [16] and for Cl from [17]. The F_v value for Cs corresponds to soil pH <4 (taiga).

Radionuclide uptake by animals and their transition to milk and meat

Radionuclide uptake by animals depends on their type, age, weight gain, digestibility of feed and, in the case of lactating animals, milk yield. This approach assumes that the feed does not contain radionuclides with solely drinking water being considered as an uptake source for the animals.

Concentration in cow milk

Radionuclide concentration in milk depends directly on its concentrations in drinking water consumed by the milk-producing animals. Concentration of the i-th radionuclide in milk $C_{m,i}$ is calculated based on its concentration in drinking water $C_{w,i}$:

$$C_{m,i} = F_m (C_{w,i} Q_w)\exp(-\lambda_i t_m), \qquad (8)$$

where $C_{m,i}$ [Bq/l] is the i-th radionuclide concentration in milk; F_m [day/l] is the fraction of activity passed from feed to milk under equilibrium conditions (specific for each chemical element); $C_{w,i}$ [Bq/m^3] is the i-th radionuclide concentration in water; Q_w [m^3/day] is the daily water consumption by animals (for lactating animals it is assumed to be equal to 0.06 m^3/day); λ_i [year^{-1}] is the radioactive decay constant for the i-th radionuclide; t_m [year] is the time period between milking and milk consumption (1 day is assumed for the fresh milk or $2.7\cdot10^{-3}$ year).

Radionuclide concentrations in cattle meat

Radionuclide concentration in meat is calculated similarly to its concentration in milk under the same restrictions:

$$C_{f,i} = F_f (C_{w,i} Q_w)\exp(-\lambda_i t_f), \qquad (9)$$

where $C_{f,i}$ [Bq/kg] is the i-th radionuclide concentration in meat; F_f [day/kg] accounts for the portion of activity that passes from feed to meat under equilibrium conditions or during slaughter (specific for each chemical element); $C_{w,i}$ [Bq/m^3] is the i-th radionuclide concentration in water; Q_w [m^3/day] is the daily water consumption by animals (assumed as 0.04 m^3/day for beef cattle); λ_i [year^{-1}] is the radioactive decay constant for the i-th radionuclide; t_f [year] is the time period between slaughter and meat consumption (assumed to be equal to 20 days or 0.055 year).

Expected concentrations in livestock products

Table 4 provides the specific values of concentration factors, as well as the expected radionuclide concentrations in milk and meat.

Table 4. Concentration factors and expected radionuclide concentrations in milk and cattle meat

Nuclide	F_m*, day/l	F_f*, (day/kg)	$C_{m,i}$, Bq/l (milk)	$C_{f,i}$, Bq/kg (meat)
^{14}C	$9.0\cdot10^{-3}$	$8.0\cdot10^{-2}$	$2.7\cdot10^{-8}$	$1.6\cdot10^{-7}$
^{36}Cl	$2.0\cdot10^{-2}$	$2.0\cdot10^{-2}$	$6.0\cdot10^{-8}$	$4.0\cdot10^{-8}$
^{79}Se	$1.0\cdot10^{-3}$	$1.0\cdot10^{-1}$	$3.0\cdot10^{-9}$	$2.0\cdot10^{-7}$
^{93}Mo	$5.0\cdot10^{-3}$	$1.0\cdot10^{-2}$	$1.5\cdot10^{-8}$	$2.0\cdot10^{-8}$
^{94}Nb	$4.0\cdot10^{-6}$	$3.0\cdot10^{-6}$	$1.2\cdot10^{-11}$	$6.0\cdot10^{-12}$
^{99}Tc	$1.0\cdot10^{-3}$	$1.0\cdot10^{-3}$	$3.0\cdot10^{-9}$	$2.0\cdot10^{-9}$
^{129}I	$1.0\cdot10^{-2}$	$5.0\cdot10^{-2}$	$3.0\cdot10^{-8}$	$1.0\cdot10^{-7}$
^{135}Cs	$1.0\cdot10^{-2}$	$5.0\cdot10^{-2}$	$3.0\cdot10^{-8}$	$1.0\cdot10^{-7}$
^{226}Ra	$1.0\cdot10^{-3}$	$5.0\cdot10^{-3}$	$3.0\cdot10^{-9}$	$1.0\cdot10^{-8}$
^{235}U	$6.0\cdot10^{-4}$	$3.0\cdot10^{-3}$	$1.8\cdot10^{-9}$	$6.0\cdot10^{-9}$
^{238}U	$6.0\cdot10^{-4}$	$3.0\cdot10^{-3}$	$1.8\cdot10^{-9}$	$6.0\cdot10^{-9}$

*No F_m and F_f values were indicated in [11] for C and Cl. Values for C were taken from [16] and for Cl from [17].

Radiation exposure of population

Under this approach, doses for a hypothetical critical group of population are simultaneously summed up over all radionuclides and all the considered exposure routes. However, in real-life this situation seems to be quite unlikely, but it can be considered as a reasonable approximation in case of typical estimates. As mentioned earlier, considered are the maximum doses for a 10,000-year long exposure from a source. The calculations are based either on the assumption of achieving equilibrium nuclide activities or accumulation of long-lived nuclides activities.

Calculation of peroral doses was performed as follows:

$$E_{ing,p} = C_{p,i} H_p DF_{ing}, \qquad (10)$$

where $E_{ing,\,p}$ is the annual effective dose from the consumption of the i-th radionuclide in the product p [Sv/year]; $C_{p,i}$ is the concentration of the i-th radionuclide in the product p [Bq/kg (l)]; H_p is p-th product consumption rate [kg/year]; DF_{ing} is the dose coefficient indicating the internal intake of the i-th radionuclide [Sv/Bq].

Tables 1 and 2 present the assumed rations and dosimetric parameters, respectively.

Tables 5 and 6 summarize the expected intake by different routes and total doses for a single radionuclide flux from DRWDF given the considered critical groups.

Table 5. Expected intake given different exposure routes and total doses for a single radionuclide flux from DRWDF (children)

Nuclide	Water (Sv/year)	Vegetables (Sv/year)	Milk (Sv/year)	Meat (Sv/year))	Total (Sv/year)
^{14}C	$2.1 \cdot 10^{-14}$	$1.7 \cdot 10^{-12}$	$7.0 \cdot 10^{-15}$	$6.1 \cdot 10^{-15}$	$1.8 \cdot 10^{-12}$
^{36}Cl	$8.2 \cdot 10^{-14}$	$9.8 \cdot 10^{-13}$	$6.2 \cdot 10^{-14}$	$6.0 \cdot 10^{-15}$	$1.1 \cdot 10^{-12}$
^{79}Se	$3.6 \cdot 10^{-13}$	$4.9 \cdot 10^{-11}$	$1.4 \cdot 10^{-14}$	$1.3 \cdot 10^{-13}$	$5.0 \cdot 10^{-11}$
^{93}Mo	$9.0 \cdot 10^{-14}$	$1.1 \cdot 10^{-11}$	$1.7 \cdot 10^{-14}$	$3.3 \cdot 10^{-15}$	$1.1 \cdot 10^{-11}$
^{94}Nb	$1.3 \cdot 10^{-13}$	$1.6 \cdot 10^{-12}$	$1.9 \cdot 10^{-17}$	$1.4 \cdot 10^{-18}$	$1.7 \cdot 10^{-12}$
^{99}Tc	$6.2 \cdot 10^{-14}$	$1.3 \cdot 10^{-13}$	$2.3 \cdot 10^{-15}$	$2.3 \cdot 10^{-16}$	$1.9 \cdot 10^{-13}$
^{129}I	$2.5 \cdot 10^{-12}$	$1.5 \cdot 10^{-12}$	$9.3 \cdot 10^{-13}$	$4.6 \cdot 10^{-13}$	$5.4 \cdot 10^{-12}$
^{135}Cs	$2.6 \cdot 10^{-14}$	$3.8 \cdot 10^{-14}$	$9.8 \cdot 10^{-15}$	$4.8 \cdot 10^{-15}$	$7.9 \cdot 10^{-14}$
^{226}Ra	$2.0 \cdot 10^{-11}$	$2.7 \cdot 10^{-10}$	$7.3 \cdot 10^{-13}$	$3.6 \cdot 10^{-13}$	$2.9 \cdot 10^{-10}$
^{235}U	$1.7 \cdot 10^{-12}$	$2.5 \cdot 10^{-11}$	$3.8 \cdot 10^{-14}$	$1.9 \cdot 10^{-14}$	$2.7 \cdot 10^{-11}$
^{238}U	$1.6 \cdot 10^{-12}$	$2.3 \cdot 10^{-11}$	$3.5 \cdot 10^{-14}$	$1.7 \cdot 10^{-14}$	$2.5 \cdot 10^{-11}$

Table 6. Expected intake given different exposure routes and total doses for a single radionuclide flux from DRWDF (adults)

Nuclide	Water (Sv/year)	Vegetables (Sv/year)	Milk (Sv/year)	Meat (Sv/year)	Total (Sv/year)
^{14}C	$1.7 \cdot 10^{-14}$	$1.7 \cdot 10^{-12}$	$2.1 \cdot 10^{-15}$	$5.5 \cdot 10^{-15}$	$1.7 \cdot 10^{-12}$
^{36}Cl	$2.8 \cdot 10^{-14}$	$3.9 \cdot 10^{-13}$	$7.6 \cdot 10^{-15}$	$2.2 \cdot 10^{-15}$	$4.3 \cdot 10^{-13}$
^{79}Se	$8.7 \cdot 10^{-14}$	$1.4 \cdot 10^{-11}$	$1.2 \cdot 10^{-15}$	$3.4 \cdot 10^{-14}$	$1.4 \cdot 10^{-11}$
^{93}Mo	$9.3 \cdot 10^{-14}$	$1.4 \cdot 10^{-11}$	$6.3 \cdot 10^{-15}$	$3.7 \cdot 10^{-15}$	$1.4 \cdot 10^{-11}$
^{94}Nb	$5.1 \cdot 10^{-14}$	$7.6 \cdot 10^{-13}$	$2.8 \cdot 10^{-18}$	$6.0 \cdot 10^{-19}$	$8.1 \cdot 10^{-13}$
^{99}Tc	$1.9 \cdot 10^{-14}$	$4.6 \cdot 10^{-14}$	$2.6 \cdot 10^{-16}$	$7.6 \cdot 10^{-17}$	$6.6 \cdot 10^{-14}$
^{129}I	$3.3 \cdot 10^{-12}$	$2.4 \cdot 10^{-12}$	$4.5 \cdot 10^{-13}$	$6.5 \cdot 10^{-13}$	$6.8 \cdot 10^{-12}$
^{135}Cs	$6.0 \cdot 10^{-14}$	$1.0 \cdot 10^{-13}$	$8.2 \cdot 10^{-15}$	$1.2 \cdot 10^{-14}$	$1.8 \cdot 10^{-13}$
^{226}Ra	$8.4 \cdot 10^{-12}$	$1.3 \cdot 10^{-10}$	$1.1 \cdot 10^{-13}$	$1.7 \cdot 10^{-13}$	$1.4 \cdot 10^{-10}$
^{235}U	$1.4 \cdot 10^{-12}$	$2.5 \cdot 10^{-11}$	$1.2 \cdot 10^{-14}$	$1.7 \cdot 10^{-14}$	$2.6 \cdot 10^{-11}$
^{238}U	$1.4 \cdot 10^{-12}$	$2.4 \cdot 10^{-11}$	$1.1 \cdot 10^{-14}$	$1.6 \cdot 10^{-14}$	$2.5 \cdot 10^{-11}$

The approach considered in the article allowing to assess the radiation exposure of population due to water use was also implemented in the Ecorad software and calculation complex. The presented results (Tables 5 and 6) were also obtained using this software product. Moreover, Ecorad calculation algorithms were built in accordance with a methodology approved by Rostekhnadzor providing for the development of standards for permissible discharges of radioactive substances [23], as well as assuming the possibility of implementing relevant approaches and databases used in the RESRAD family of program codes [24].

Conclusions

Conversion coefficients (CC) enabling to establish relationship between the activity of a radionuclide flow from deep RW disposal facilities into biosphere (Bq/year) and the expected radiation exposure (Sv/year) were derived in this study, namely, for a scenario suggesting contaminated water use for drinking purposes, as well as cattle watering and irrigation of farm households where products are grown for own consumption.

Tables 5 and 6 demonstrate that the main dose contribution is associated with the consumption of vegetables being watered with contaminated water. Vegetable contamination is mainly due to root uptake[2]. This fact is considered as a distinctive feature of the radiation exposure associated with DRWDF and relevant biospheric coefficients. Length of the source exposure (indicated as t_b in formula (4)) is

[2]Apparently, conservatism of the model applied is associated precisely with the factor Fv indicating the way in which the radionuclides are concentrated from the soil into plants. Values of this parameter are more than an order of magnitude higher than those used in more realistic models.

assumed to be equal to 10,000 years, whereas for "common" nuclear power facilities the characteristic value of t_b in [11] accounts for 30 years. For this reason, the exponential factor $\left[1-\exp(-\lambda_{E_i^s} t_b)\right] / \lambda_{E_i^s}$ taking into account the correction for achieving equilibrium radionuclide activity in the biosphere depending on the characteristic values of $\lambda_{E_i^s}$ can increase by several orders of magnitude. This fact highlights the accumulation of long-lived radionuclides in the soil and the predominance of radiation exposure during the consumption of contaminated plant products compared to drinking water. For "common" nuclear power facilities, a reverse trend is usually observed.

A comparison of CC for two critical groups, "adults" and "children", shows that they lie within the same order — the maximum $CC_{children}/CC_{adults}$ ratios are observed for ^{79}Se (3.6), ^{36}Cl (2.6), ^{99}Tc (2.9), whereas for ^{135}Cs this ratio accounts for 0.4. These differences are much smaller than the accuracy of the models applied, and for future assessments it seems reasonable to consider if among the population only one critical group (adults) should be distinguished. Moreover, only one critical group is commonly used in safety assessments abroad which is explained by the same reasons.

Table 7 provides a comparison of the calculated CCs for a critical group of adults with the CCs used in some international safety assessments for the considered radionuclides. It should be noted that in the international safety assessments acceptable risk CCs values were given (10^{-6} year for the considered operations). The CC values were recalculated based on the values of linear radiation risk coefficients provided in relevant studies. These coefficients relate the risk of malignant neoplasms and hereditary defects per an effective dose unit.

Table 7. Comparison of CC from this study (adults) and from international safety assessments (Sv/Bq)

Nuclide	Present study	[16] UK	[18] UK	[19] Finland	20] Sweden
^{14}C	$1.7\cdot10^{-12}$	$1.02\cdot10^{-12}$	$1.21\cdot10^{-11}$	$3.33\cdot10^{-13}$	$5.4\cdot10^{-12}$
^{36}Cl	$4.3\cdot10^{-13}$	$1.29\cdot10^{-13}$	$1.49\cdot10^{-12}$	$3.33\cdot10^{-13}$	$5.8\cdot10^{-13}$
^{79}Se	$1.4\cdot10^{-11}$	$1.4\cdot10^{-12}$	$1.64\cdot10^{-11}$	$1\cdot10^{-12}$	$1.2\cdot10^{-9}$
^{93}Mo	$1.4\cdot10^{-11}$	$1.73\cdot10^{-13}$	$1.72\cdot10^{-12}$	$3.33\cdot10^{-14}$	$9.0\cdot10^{-13}$
^{94}Nb	$8.1\cdot10^{-13}$	$2.99\cdot10^{-12}$	$3.47\cdot10^{-11}$	$1\cdot10^{-12}$	$4.0\cdot10^{-12}$
^{99}Tc	$6.6\cdot10^{-14}$	$2.03\cdot10^{-14}$	$2.08\cdot10^{-13}$	$3.33\cdot10^{-14}$	$9.0\cdot10^{-13}$
^{129}I	$6.8\cdot10^{-12}$	$3.24\cdot10^{-12}$	$3.27\cdot10^{-11}$	$1\cdot10^{-12}$	$6.5\cdot10^{-10}$
^{135}Cs	$1.8\cdot10^{-13}$	$4.97\cdot10^{-13}$	$5.79\cdot10^{-12}$	$3.33\cdot10^{-13}$	$4.0\cdot10^{-14}$
^{226}Ra	$1.4\cdot10^{-10}$	$4.89\cdot10^{-11}$	$5.21\cdot10^{-10}$	$3.33\cdot10^{-12}$	$3.8\cdot10^{-12}$
^{235}U	$2.6\cdot10^{-11}$	$1.4\cdot10^{-12}$	$1.40\cdot10^{-11}$	$3.33\cdot10^{-12}$	$1.9\cdot10^{-12}$
^{238}U	$2.5\cdot10^{-11}$	$1.37\cdot10^{-12}$	$1.37\cdot10^{-11}$	$3.33\cdot10^{-12}$	$1.9\cdot10^{-12}$

Radiation effects in [16, 18—20] were calculated using more realistic models accounting for the setup at particular DRWDF sites with the landscape parameters differing from those assumed in [10—11]. However, being obtained using different methods, CC values basically differ within an order of magnitude only.

It should be noted that in addition to using a simplified conservative model [11], conservatism of the estimates also results from an assumption suggesting that the contaminated underground flow is discharged into the surface layer of sedimentary rocks. The most likely ground water discharge way for the DRWDF case could be its discharge into underflow area of a river [14, 21]. Preliminary estimates show that CC in this case becomes several orders of magnitude lower [22] due to large dilution in the river channel (water flow in the section for a large river can be several thousand m^3/sec).

As mentioned before, the above approach is simplified and conservative and can only be used for preliminary assessments. If under this approach the DRWDF radiation exposure appears to be acceptable, then the safety of the facility is very likely to be ensured. On the contrary, negative results obtained only indicate the necessity of conducting more realistic research and acquiring additional input data.

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